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A REVIEW OF MONITORING LOW DENSITY ANIMAL POPULATIONS

Jennifer Brown

Biomathematics Research Centre
Department of Mathematics and Statistics
University of Canterbury
Private Bag 4800
Christchurch, New Zealand
Email: j.brown@math.canterbury.ac.nz

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1. SOME GENERAL PRINCIPLES RELEVANT TO ALL MONITORING PROGRAMMES

1.1 MONITORING OBJECTIVES

One of the most important steps in designing a monitoring programme is to have very clear objectives. The objectives should define what is the state of the environment desired by managers, over what time frame, in what spatial location, and at what spatial scale. Clear objectives can help define what is the being estimated - change over time in numbers, or in spatial coverage of the species. Objective targets need to be measurable and quantified. The need to define objectives is discussed in a very readable paper by Gibbs *et al.* (1999) where they talk about effective monitoring for adaptive wildlife management. They argue that the effectiveness of monitoring, in terms of how well the results feed into management practice, is determined by how well the objectives of the monitoring are set. In particular it is important to have objectives that, "...specifically describe some desired state of an appropriate indicator that management is intended to met..."

1.2 ERRORS ASSOCIATED WITH SAMPLING

Monitoring is usually undertaken by sampling where only a fraction of the population is measured. The survey results may not reflect the true situation and there may be more or less animals than the results suggest. It is important to have a good understanding of sampling errors. A description written for DOC of type I (α) and II (β) errors is given in Brown and Miller (1998). As a brief refresher, α is the chance of falsely rejecting the null hypothesis (i.e., saying there is a change or a trend when there is not) and β is the chance of falsely accepting the null hypothesis (i.e., saying there is no change when there is). Power is the chance of correctly detecting a change, and is calculated as $1-\beta$. There are many general references for conservation biologists (e.g., Peterman 1990, Fairweather 1991, Taylor and Gerrodette 1993, Steidl *et al.* 1997) on power, and how to set the acceptable levels for α and β.

In biological conservation high α values, e.g., 0.1 or 0.2 are often recommended to ensure smaller β. However, for pest monitoring to detect a decline after control a small α may be preferable to a small β. Falsely concluding there has been a decline in the pest population would be more damaging for the conservation resource than incorrectly concluding an operation has not been successful. The decision on acceptable levels of these two errors should include consideration of costs - what is the cost of making the wrong decision either way. The ratio between these two costs can be used to set the acceptable error levels (Mapstone 1995). An example of this is given in Brown and Miller (1998 p.18.) where as a first estimate the ratio of the cost of β to α was set at 0.5, "...i.e., the immediate short term loss of falsely concluding a control <operation> was successful when it was not, is twice the long term cost of failing to detect a successful control. This is because if a control were not successful managers would need to undertake extra protection work for the conservation resources. If the unsuccessful control operation were thought to be successful extra work would not be carried out." With a cost ratio of 0.5, the ratio between α and β should be 0.5. If α is set to 0.1 then β should be 0.2, and power will be 0.8.

Discussion on power and α and β can led one to think that the true power and error levels are known prior to sampling. The true power and error levels are not known and are only *estimated* from calculations that usually involve estimates of variation. The importance of understanding statistical power is more in understanding of the sources of sampling variation and for assisting in designing an efficient monitoring programme than in the actual values calculated.

Variation in data collected in monitoring studies is due to:
i) Within-site variation which reflects the inexactness of the data collection,
ii) The variation among sites due to the environmental heterogeneity, and
iii) Temporal variation.

(Millard and Lettenmaier 1986, Gerrodette 1987, Link et al. 1994, Brown and Manly 2001, p. 9 module 3). Power will improve if any of these sources of variation are reduced. The optimal design can be thought of as where these sources of variation have been quantified and the correct trade-off made between e.g., allocating effort into a site to reduce within-site variation and allocating effort into visiting many sites to reduce among site variation.

As a general statement, for trend detection, having more sample units is generally preferable to increasing effort within a unit (Millard and Lettenmaier 1986, Link *et al.* 1994, Van der Meer 1997, Brown and Manly 2001, p. 9 module 3). For example, for monitoring stoats Brown and Miller (1998) suggest having more tracking tunnels revisited a few times rather than a few tunnels that are revisited many times, and for monitoring possums Brown and Thomas (2000) show that having many shorter lines of leg-hold traps is preferable to having a few long lines. (In the possum study the line of traps is considered a sample unit).

Another design consideration that can affect the power for detecting trends is whether to revisit the same location of the sample unit for each survey, or to visit new locations. In general, when the interest is in a specific trend, or change, at a locality then revisiting the same location (e.g., re-measure the same quadrat) will be more powerful than selecting new units at each survey occasion (Skalski 1990, Van der Meer 1997, Brown and Manly (2001, p. 4 module 3). This is because revisiting the same site should remove any confounding effect of spatial variation, and any differences seen should be due to temporal changes. If, on the other hand, the interest is in status at each time, (e.g., what is the average density of the plant at time x and the average density at time y) then reselecting (randomly) chosen units is recommended. Other options are to revisit some of the units and, at each sample occasion, increase spatial coverage by introducing new units (and dropping others out) (Skalski 1990, Urquart *et al.* 1993). Such an approach is called sampling with partial replacement and some of these designs are discussed in Brown and Manly (2001, p. 4 module 3).

2. CHALLENGES WITH MONITORING LOW DENSITIES

Some of the challenges with monitoring low-density populations are that when the population is at low numbers:

i) Sampling error may mean that animals are present but not detected,
ii) Counts of animals are discrete numbers and the detection of one additional animal can have an unduly large effect on the estimate of abundance, and
iii) Commonly used analysis methods to estimate abundance may not be robust to samples with many zero counts.

The first of these, animals are present but not detected, means that there is uncertainty that if animals are not detected at a site, or sample unit, there is some uncertainty that the unit is truly unoccupied. There are various methods to deal with this uncertainty. One way to minimize the uncertainty is by devising a sampling scheme that reduces the chances of failing to detect animals by allocating effort to surveying intensively within each unit (e.g., by using many sample devices within the unit, or many observers), or by using very small unit areas. Another way is to model the uncertainty by estimating the detection probability for each unit area (MacKenzie *et al.* 2001).

An example of the consequence of small changes in the counts of observed animals (i.e., discrete numbers) is given in Brown and Thomas (2000). At high densities catching one more possum has little effect on the standard possum-monitoring index, residual trap catch (RTC). For example, at 20% RTC one additional possum would give an RTC of 20.33% (or a 1.67% increase). At low densities, say 2% RTC, one additional possum would give an RTC of 2.33% (or a 16.67% increase). The consequence of catching an additional possum at high densities is less than at low densities. The low-density RTC estimate is more sensitive to small differences in trap-catch rates (Brown and Thomas 2000). This may seem rather esoteric, but in reality when important management decisions are being made on the basis of indices (e.g., whether to pay a contractor for their control work) then the efficacy of such indices at low densities should be questioned. In general survey and analysis methods should be used that are not sensitive to such sampling errors, e.g., methods based on presence/absence rather than counts of individuals.

The third point listed is the challenge of analysing data from monitoring low densities. Data from counts is typically highly skewed and contains many zero counts. Standard analysis methods, such as confidence intervals based on the empirical formulae (estimate $\pm t \cdot$ standard error) can give misleading results. For example, Brown and Thomas (2000) found that the empirical method to estimate 95% confidence intervals for possum RTC typically only provided 90% coverage. Various alternative methods exist for estimating confidence intervals from skewed data (Manly 1997, p. 34 - 59) and specifically for possum RTC (Brown and Thomas 2000, Webster and Caley 2001). Again, appropriate analysis methods should be used for data from low-density surveys.

3. AN APPROACH FOR MONITORING LOW DENSITY POPULATIONS

3.1 SURVEYING FOR SITE OCCUPANCY

An approach for monitoring low-density populations is to focus on detecting the proportion of the study area that has the species present rather than aiming to estimate density or abundance. The information from this approach can be used to estimate the spatial distribution of the population within the study area.

The simplest method for this approach is to count the proportion of unit areas that have the species present. These unit areas are the sample units and can be of any size depending on the size of the study area and the species. As an example, a 1000ha study area could be divided into equal 1 ha unit areas equating to a grid cell on a topographical map. The estimate of the proportion of the areas occupied is then

$$\hat{\psi} = \frac{N'}{N}$$

where there are N' units of the N units occupied. Usually N' is estimated and hence ψ is an estimate only.

The idea of presence/absence surveys is intuitively appealing for low density animal monitoring. It is often logistically easier to detect presence in a unit area than to count the number of animals in the area. Further, if animals are spatially aggregated (at the scale of the unit areas) then, when one animal is detected it is likely another will be detected in the same unit. Counting the additional animal in the unit areas is not providing as much new information as detecting the first animal. For example, there is evidence that catches of possums in traps are correlated - when one trap catches a possum it is likely the neighbouring trap will also catch a possum (Brown and Thomas 2000, Faddy *et al.* 2001). Therefore the difference between catching one or two possums on a trap line is of less interest to the difference between catching zero or one possum (assuming a zero catch equates to an absence of possums).

The advantage of surveying for presence/absence is that devices that merely record the presence of animals can be used in preference to devices that count the number of animals. Devices that record animal presence can be lightweight and so many more can be used compared with counting-devices. There are many alternative devices being developed by industry including hair traps, wax blocks, long-life ink blocks for tracking tunnels (e.g., Pest Control Research Ltd). Lindenmayer *et al.* (1999) have reviewed some of the alternative hair traps for Australian situation.

3.2 DESIGNS FOR PRESENCE ABSENCE SURVEYS

The sample units in a presence/absence survey are the N unit areas. The number of unit areas surveyed is n. The estimate of occupancy rate is based on the proportion of these occupied. Determining the proportion of the study area that contains animals is straightforward if presence (or more correctly absence) is known with certainty. If an animal is detected then there is 100% certainty that the unit is occupied. The opposite is not as straightforward. If no animal is detected there is some uncertainty about whether the unit is truly unoccupied.

Designs that have highly intensive surveys within units will minimize the chance of failing to detect the unit is occupied, but fewer units can be surveyed for a fixed amount of effort. There will be an optimal balance between:

i) Surveying a small proportion of the N units with more effort within each unit. The probability that a present species is detected will be high but only a few units will have been surveyed.

ii) Surveying a large proportion of the N units with minimal effort within each unit. The probability that a present species is detected will be low but many units will have been surveyed.

3.3 SURVEY EFFORT WITHIN UNIT AREAS

Survey effort within each unit can be increased by: multiple detection devices, multiple surveys to the unit, or from multiple observers (Nicholas and Karanth 2001). With multiple observers the estimate of the detection probability, and N', can be by using a Lincoln-Petersen estimator. With multiple visits N' can be estimated from the data from the sequence of site visits. For example, if a unit were visited six times, and animals detected on the 4th and 6th visits, the data recorded for that unit would be 0 0 0 1 0 1. Visits 1, 2, 3 and 5 recorded no animals, and visits 4 and 6 recorded at least one animal. Removal or mark-recapture methods (MacKenzie *et al.* 2001, Nicholas and Karanth 2001) can be used to estimate the unit's detection probabilities. This approach seems very promising and should be explored further.

A simpler (but possibly less precise) approach for reducing uncertainty from failing to detect that a unit is occupied is to use a sequential sampling scheme. For example Brown and Boyce (1996) recommended for monitoring butterflies multiple visits to each survey unit until butterflies are detected, up to a maximum of three visits. If butterflies were detected at any of these visits then the unit was obviously occupied. If after three visits no butterflies were detected the unit was defined to be unoccupied. The choice of three visits was based on a simple binomial model. For the defined level of survey effort their study indicated there was an 80% chance of detecting at least one butterfly given that butterflies were present. Three visits to the site reduced the chance of failing to detect butterflies in occupied sites to a minimum. To estimate the chance of detecting at least one butterfly in an occupied site they conducted a number of very intensive surveys and modelled detection rates with lower levels of survey effort. This approach is simplistic because it assumes constant detection probabilities among all units regardless of the number of butterflies present in the units.

The difference between the sequential sampling method used by Brown and Boyce (1996) and the methods that estimate detection probability is that in the Brown and Boyce method the resultant data for each unit approach is binary - 1 if butterflies were detected and 0 if not, whereas when detection probability are estimated the data is a measure of the likelihood animals were present. With both methods N' can be estimate and then ψ estimated but presumably simple sequential sampling method will be less precise.

A practical difference between these two methods is that when mark-recapture methods are used, units are visited a fixed number of times regardless of whether animals are detected because the sequence of data on whether the animal was detected or not is used in the analysis. With the sequential design a unit need only be visited until the animal is detected (or until the maximum number of visits is reached). If an animal is detected on one of the initial site visits, survey effort can then be reallocated to sampling additional units.

3.4 SIZE OF UNIT AREA SIZE AND NUMBER OF UNIT AREAS

The choice of the number of sample units, depending on the design and analysis method, can be based on some power analysis.

In a simple design where a selection of unit areas are visited once, Green and Young (1993) present an eloquent argument that when populations are rare the distribution of animals/unit can be adequately described by a Poisson model. They define rare as densities less than 0.1 animals per unit area. Then, the estimate of the number of sample units to detect the presence of an animal/s in at least one of the unit areas (i.e., to estimate if the animal is present or not in the entire study area) is simply $n = -1 \cdot (1/m) ln\beta$, where β is the type II error and m is the mean density of the species. The concept of rarity is relative to the unit area. If animal numbers are higher than $m = 0.1$ then smaller unit areas will reduce the density.

For more sophisticated approaches, e.g., where the detection probabilities are being estimated from a model as in the MacKenzie *et al.* (2001) method than the minimum sample size needs to be quantified in more complex manner. This is an area of continuing research.

The issue of what is the appropriate size for the unit area has been addressed in various papers, given the general view that many small sample units are preferable to a few large units. An issue with presence/absence surveys is that they can underestimate the rate of decline (or increase) in a species. With large unit areas a species would have to decline to low levels before a unit would be declared unoccupied. For example, a 89% decline in the population of a butterfly species (*Lycaean phlaeas*) would have only been recorded as a 15% decline if monitoring had used presence/absence counts in 1 km^2 units, or a 31% decline if 500 m^2 units had been used (Thomas and Abery 1995, León Coretés *et al.*, 1999, 2000).

3.5 ENHANCEMENT OF SURVEY DESIGNS

So far the simplest survey design has been considered - a simple random sample of unit areas. This design can be improved by using a stratified design where strata are created from areas of similar density, or habitat type (Lohr 2000). The usual approach in stratified sampling is to allocate more sampling effort (i.e., to sample more of the unit areas) in the more variable stratum, but there may be some advantages in allocating more effort into low-density strata (Brown and Miller, Zeilinski and Stauffer 1996). If a modeling approach were used then another approach to deal with habitat and/or density differences would be to use habitat-covariates in the analysis. If there were information on habitat then yet another approach is to use these habitat data alone to predict the species distribution. This was done successfully for butterflies in one study (Cowley *et al.* 2000). Whether this approach could be used as successfully for other animals would depend on the robustness of the habitat data/animal presence model.

Another consideration would be to use an adaptive design rather than simple random sampling (either over the whole study area of within strata). Adaptive sampling is where information gained during the survey is used to adapt the sampling plan (Thompson and Seber 1996). For example, the survey plan could involve surveying units surrounding any unit where animals are detected. This is called adaptive cluster sampling (Thompson 1990, Brown 1996). Other survey designs based on unequal probability of selection could also be useful to consider. For example, an unequal probability of selection design could allow for greater chance of selecting sample units that contain large amounts of preferred habitat. In this example the units surveyed would be the ones most likely to be occupied by the species of interest. Systematic sampling should also be considered. The advantage of systematic sampling is that the design can ensure spatial coverage over the entire study area (Brown and Thomas 2000).

There is some interest in considering ecological processes in a spatial context, e.g., to view the density of animals as a continuous surface on the domain of the population (Stevens 1997). In the context of this review, rather than estimating the proportion of occupied unit areas, the interest would be in the spatial arrangement of the units where each unit has a probability of being occupied. This is targeted as an area for further work by the authors of MacKenzie *et al.* (2001). Regardless of how sophisticated the survey design and analysis the survey data should be stored in a format that maintains the spatial information, i.e., using a map to mark which units were surveyed and site occupancy information and using a GIS system. Information on spatial distribution is very useful for managers. For example, the identification of local residual patches of possums within a large block can be used to target follow-up control.

A good review of the use of spatial models for wildlife is given in Buckland and Elston (1993). The most widely used geostatistic methods are based on krigging. Krigging is used for prediction of processes in space based on the observations at known points and the correlation among these points. The method allows prediction in areas where no data has been observed. The classic paper on spatial correlation for ecologists is Legendre (1993). Both this paper and the Buckland and Elston (1993) paper are mentioned here for completeness but not reviewed.

As a final point, presence/absence approach has been used in other applications. For example, in horticulture commercial crops are sampled to detect the presence of disease or infestation. Such schemes can be sequential where the sampling of e.g., leaves continues up to a fixed maximum, although effective non-sequential schemes have been proposed (Hepworth and MacFarlane 1992). The difference here between sampling for presence of insect damage and sampling to detect presence of animals is that there is more certainty in the commercial crop situation that failure to detect sign is evidence of absence of the insect. In the animal situation failure to detect sign in a unit area may not be strong evidence of absence. Some of the entomological literature has been reviewed, e.g., some butterfly papers, but the full range has not been covered.

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